Post-TCRA risks were evaluated for dioxins and furans only. Data or representative concentrations for all COPCHS in all media of interest for post-TCRA conditions were not available and, therefore, dioxins and furans were used to provide a relative measure of hazard and/or risk. EPCs representative of post-TCRA conditions for each medium were estimated using a variety of methods. For sediments and soils, the portion of the baseline data from within the exposure units defined for the post-TCRA condition (i.e., defined as the areas that were still accessible to individuals following the TCRA) were used. No tissue data were collected following the TCRA. In the absence of such data, post-TCRA tissue concentrations for hardhead catfish were estimated using statistical relationships between baseline sediment and tissue samples established in the *Technical Memorandum on Bioaccumulation Modeling* (Integral 2010c). For clams and crabs, where no meaningful model for predicting sediment–tissue relationships existed, assumptions regarding the baseline dataset were used to estimate post-TCRA EPCs. Appendix F documents the detailed methods used for post-TCRA EPCs as well as the post-TCRA risk characterization results and the uncertainties associated with these estimates.

5.1.2.2.2 Exposure Parameters

This section provides an overview of the exposure assumptions used in the deterministic evaluation. A detailed presentation and the supporting rationales for these assumptions are included in the EAM (Appendix A). A summary of these exposure parameters is presented in Table 5-6. Assumptions adopted for chemical specific exposure parameters are provided in Table 5-7.

Differences in activity and intake parameters have been characterized for younger children, older children, and adults. Therefore, exposure parameters were developed separately for young children (ages 1 to <7 years), older children (ages 7 to <18 years), and adults (ages 18 years and older).

Considering the exposure factors assumed for this BHHRA, young children would have higher potential exposures (on a per unit body weight basis) relative to other age groups. Therefore, for the RME scenarios for all human receptor groups evaluated, it was assumed that a portion of the total exposure occurs at these younger life stages. This is a conservative assumption because it results in an upper-bound RME scenario in which the calculated exposure for any alternative age grouping over the same chronic exposure duration would be lower. As established in the EAM, the individuals considered most likely to use the area under study under baseline conditions are adults. Therefore, for the CTE analysis, only adult exposures are evaluated. It is however, recognized that children may frequent the area along with adults. At the request of USEPA (comment 9 of the draft BHHRA, Appendix N) an additional CTE analysis for a hypothetical young child receptor was performed and is presented in the uncertainty evaluation.

Common Parameters

Given the lack of specific information on fishing and recreational behaviors within USEPA's Preliminary Site Perimeter, the exposure durations were conservatively based upon standard default assumptions used for residents. Default exposure durations of 33 years for the RME and 12 years for the CTE (USEPA 2011a) were based on studies of occupational mobility, and were adopted for this BHHRA.

Following common practice for human health risk assessment, the averaging time selected depended on the toxic endpoint (cancer or noncancer) being assessed. For noncarcinogens, the averaging time was set equal to the exposure duration (e.g., for an exposure duration of 6 years the averaging time was 2,190 days). For carcinogens that were evaluated with a CSF, the averaging time was set equal to a lifetime (i.e., 78 years or 28,470 days) (USEPA 1989, 2011a). When the toxicity of a carcinogen was described using a criterion that assumed a threshold dose was required for an adverse effect to be elicited (i.e., TEQ_{DF}) the averaging time was set equal to the exposure duration. This latter approach described for threshold based carcinogens is essentially the same as the approach used for evaluating noncancer endpoints.

For the deterministic evaluation, mean body weights of 19, 50, and 80 kg were selected for the young child, older child, and adult age groups, respectively. These body weights were based on data collected from the 1999–2006, National Health and Nutrition Examination Survey (NHANES), and recommended in USEPA's Exposure Factors Handbook (2011a).

Parameters for Tissue Ingestion

Assumed fish and shellfish ingestion rates were selected from a study of fishing activity and consumption conducted in Lavaca Bay, Texas (Alcoa 1998). Lavaca Bay, which covers roughly 40,000 acres, is part of the larger Matagorda Bay system. This system is similar in size to Galveston Bay and is situated further south along the Texas coastline. The demographics in the counties surrounding the two bays are similar (2010 Census data for Calhoun, Chambers, Galveston, Harris, Jackson, and Victoria counties).[22]

The Lavaca Bay study collected data about consumption rates, fraction ingested from a contaminated source area, and the species composition of the fish consumed. The study was conducted during the month of November, which was reported to be the month of highest fishing activity in the bay (Alcoa 1998) and nearly 2,000 anglers participated in the study. It was conducted for the specific use of supporting a risk assessment for the Alcoa Point Comfort/Lavaca Bay Superfund Site.

Lavaca Bay ingestion rates reported by Alcoa (1998) for finfish and shellfish were adopted for this BHHRA. They were selected because they are Texas-specific and represent consumption from a fishery that is similar to the fishery associated with the area inside USEPA's Preliminary Site Perimeter. For the hypothetical recreational fisher, mean rates were used for the CTE analysis, while the 95UCL rates were used for the RME analysis. Although the Lavaca Bay study did not identify a true subsistence population for that area, the study did present upper bound (90^{th} or 95^{th} percentile) estimates of ingestion rates for the surveyed groups. These rates were selected as RME ingestion rates for the hypothetical subsistence fisher. For each of these, the average of rates for men and women were assumed for the adult ingestion rates. The rates provided for youths in the study were used to evaluate the older child while the rates provided for small children were used to evaluate exposures to the young child. The exposure frequency for ingestion of tissue was assumed to be 365 days/year for all hypothetical fishers since the fish ingestion rates used were annualized average daily averages.

[22] http://factfinder2.census.gov/faces/nav/jsf/pages/index.xhtml

Given the relatively small spatial extent of the area within USEPA's Preliminary Site Perimeter compared to the size of the Galveston Bay fishery, it is unlikely that 100 percent of the fish consumed over the 33-year-exposure duration assumed for the RME would be harvested from the area of study. The survey conducted by Alcoa (1998) at Lavaca Bay segregated the consumption data by the areas fished; specifically, a 1,500-acre subarea (indicated as the closure area), other portions of Lavaca Bay, and areas outside of Lavaca Bay. Similar to conditions at Lavaca Bay, the waters associated with USEPA's Preliminary Site Perimeter represent a very small fraction of the Galveston Bay fishery. Also like Lavaca Bay, there are many other locations around Galveston Bay that can be used for fishing. Therefore, the data from the Lavaca Bay survey were informative for the purposes of this BHHRA.

It was assumed that 25 percent of the total fish consumed by RME hypothetical recreational fishers, and 10 percent of total fish consumed by CTE hypothetical recreational fishers were collected from within USEPA's Preliminary Site Perimeter. These values were applied for the fractional intake term (FI_{tissue}) for hypothetical recreational fishers in Equation 5-4, above. Their selection is conservative for this BHHRA, as less than one percent of the fish and shellfish consumed in Lavaca Bay was from the 1,500 acre sub-area being evaluated. A full discussion of the findings of the study is found in the EAM (Appendix A).

There was no information specific to the area within USEPA's Preliminary Site Perimeter available with which to estimate the fraction intake term (FI_{tissue}) in Equation 5-4, above, for the hypothetical subsistence fisher. If subsistence activities did occur in this area, it is possible that fishers participating in these activities could fish exclusively from the waters adjacent to the area. Given the lack of information specific to fishing behaviors in the area of study, a conservative fractional intake of 1.0 was adopted for the subsistence fisher scenario.

Parameters for Direct Contact

The majority of activity by a fisher was expected and assumed to occur along the water's edge so that substantial exposure to soil was not likely. Therefore, for the fishing scenarios, the fraction of total intake that was attributed to such soils was assumed to be zero, while the fraction of total daily intake from sediment was assumed to be 1.0 (100 percent). It was envisioned, however, that the recreational visitor who is not fishing might spend equal amounts of time in contact with soils and sediments. Therefore, the fraction of total exposures attributed to soils and sediments were both assumed to be 0.5 (50 percent).

Based on USEPA's (2011a) recommended ingestion rates for soil, soil and sediment ingestion rates of 20 mg/day were assumed for adults and used to evaluate both CTE and RME estimates. An ingestion rate of 50 mg/day was assumed for older children. For younger children, a rate of 125 mg/day was assumed.[23]

For the skin surface area parameter, surface areas of 6,080 and 4,270 cm^2 were assumed for the older child and adult, respectively (USEPA 2011a), based on the assumption that an individual's hands, forearms, lower legs, and feet may come into contact with soil and/or sediment. For young children playing in the soil and/or sediment, it was assumed that the entire surface area of the leg might be in contact with sediments in addition to the hands, forearms, and feet. Based on this assumption for the young child, a surface area of 3,280 cm^2 was used (USEPA 2011a). The same surface areas were used to evaluate both the CTE and RME conditions.

Following USEPA recommendations, weighted adherence factors were calculated for each age group. These were based on the surface areas of the assumed, exposed body parts and body-part-specific adherence factors presented by USEPA (2011a) that were based on studies completed in sediment, and soil.

For sediment exposure estimates, weighted adherence factors of 3.6, 5.1, and 4.9 mg/cm^2 for young children, older children, and adults, respectively, were derived based on a study of children playing in sediment. The study was recommended by USEPA (2011a) and was one of the only available studies that investigated sediment adherence to skin. Given the difference in sediment types within USEPA's Preliminary Site Perimeter compared to those present in the study used to develop the factors presented in USEPA (2011a), and the importance of sediment type in predicting soil adherence (Spalt et al. 2009), uncertainty was

[23] Rates for the older child and young child are for the RME scenario. No child component was considered in the CTE scenario for the hypothetical recreational fisher and visitor. No CTE evaluation was completed for the hypothetical subsistence fisher scenarios.

introduced in the exposure estimates by the use of this factor. This uncertainty is further discussed within the uncertainty evaluation of the risk characterization.

A weighted soil adherence factor of 0.07 mg/cm^2 was calculated for older children and adults using data that described the adherence of soils to skin in adults participating in a variety of activities (USEPA 2011a). Data from a study conducted in children exposed to soil were used to derive a soil adherence factor of 0.09 mg/cm^2 for young children (USEPA 2011a).

The assumed exposure frequency for the direct contact pathways was based on estimates of the number of trips to the area within USEPA's Preliminary Site Perimeter each year. According to the 2006 survey of Texas anglers conducted by the U.S. Fish and Wildlife Service (USFWS), the mean number of days spent fishing marine waters by Texas residents was 13 days/year (USFWS 2006). This value was assumed for the CTE exposure frequency for direct contact pathways for the hypothetical recreational fisher. It is reasonable to assume that more avid anglers may fish with a higher frequency than the average. A survey of Maine's freshwater anglers (Ebert et al. 1993), found that the 95th percentile frequency of fishing trips per year was nearly three times that of the average number of fishing trips per year. Based on this relationship, an RME frequency of 39 days/year was assumed for the hypothetical recreational fisher. It is reasonably anticipated that hypothetical subsistence fishers, if present, may participate in fishing activities more often than recreational fishers; however, it is not likely that they would fish the same location more than an average of 2 days per week, every week of the year, over the entire assumed exposure duration of 33 years. Thus, an RME exposure frequency for direct contact pathways of 104 days/year was assumed for the hypothetical subsistence fisher scenario.

In the absence of data concerning recreational use of the area within USEPA's Preliminary Site Perimeter, RME and CTE frequencies of 104 and 52 days per year, respectively, were assumed for hypothetical recreational visitors. These were based on assumed average frequencies of 2 days per week and 1 day per week throughout the course of the year, respectively.

It is not anticipated that a fisher's or a visitor's direct contact with soils and/or sediments would typically be limited to the area within USEPA's Preliminary Site Perimeter. These individuals would likely not spend the entire day on each day that they fish or visit within this area; rather they might spend only a few hours and spend the remainder of those days engaged in activities in other areas where they could be exposed to soils or sediment from areas outside of USEPA's Preliminary Site Perimeter. No information specific to the area of study is available with which to estimate the fractional intake term for soil/sediment ($FI_{soil\text{-}sed}$) in Equation 5-1, above. Based on best professional judgment, a conservative fractional intake of 1.0 was adopted for the RME hypothetical recreational fisher and recreational visitor scenarios, and for the hypothetical subsistence fisher scenario. A fractional intake of 0.5 was adopted for the CTE scenario evaluated for the hypothetical recreational fisher and recreational visitor populations.

Chemical-Specific Factors

In addition to the scenario-specific exposure assumptions described above, there are a number of chemical-specific factors that were required to estimate $COPC_H$-specific exposure levels. These included oral bioavailability factors, dermal absorption factors, and reductions in chemical concentrations of certain $COPC_{HS}$ due to preparation and cooking. The chemical-specific values used are summarized in Table 5-7 and are briefly discussed below. A more comprehensive discussion of these parameters and the rationales for the values selected were included in the EAM (Appendix A).

Relative Oral Bioavailability

Bioavailability refers to the degree to which a substance becomes available to the target tissue after administration or exposure (USEPA 2012c). Relative bioavailability is a measure of the extent of absorption that occurs for different forms of the same chemical, different dosing vehicles, or different dose levels. Relative bioavailability adjustment (RBA) factors for oral pathways are used to account for the differences in chemical bioavailability in specific exposure media (i.e., soil, sediment, tissue) compared to the dosing vehicle used in the critical toxicity study that provides the basis for the $COPC_H$-specific toxicity criteria selected for use in this BHHRA.

The RBA can be expressed as:

$$RBA = \frac{absorbed\ fraction\ from\ exposure\ medium\ on\ site}{absorbed\ fraction\ from\ dosing\ medium\ used\ in\ toxicity\ study} \quad \text{(Eq. 5-5)}$$

In the absence of data from peer-reviewed publications or site-specific data on bioavailability of chemicals in sediment, USEPA and the Interstate and Technology Regulatory Council recommend that default factors for soil be adopted to evaluate sediment exposures (USEPA 2004; ITRC 2011). Sufficient data to determine $RBA_{soil\text{-}sediment}$ were available for dioxins and furans and for arsenic and these are discussed below. These chemical-specific RBAs were applied to the calculation of exposures via incidental ingestion of soil and sediment.

An $RBA_{soil\text{-}sediment}$ of 0.50 was adopted for dioxins and furans. This value was derived from data on the bioavailability of TCDD in soils from a range of studies selected and presented by USEPA (2010d) in their *Final Report on Bioavailability of Dioxins and Dioxin-Like Compounds in Soil*. In their report, USEPA identified six studies that reported a total of 17 RBA test results for 2,3,7,8-TCDD in soil and sediment at concentrations ranging from 1.9 to 2,300 pg/kg. These studies reported bioavailability ranging from less than 0.01 to 0.49 (i.e., <1–49 percent). The arithmetic average of the mean bioavailability from each study was 0.23 (i.e., 23 percent). This value represents the "absorbed fraction from exposure medium on site" in Equation 5-5, above, and was divided by the assumed absorbed fraction of 0.50 (i.e., 50 percent) used in establishing toxicity criteria for DLCs adopted for this BHHRA (JECFA 2002). The resulting $RBA_{soil\text{-}sediment}$ was 0.50, and this value was applied to calculation of exposures to all dioxin and furan congeners via incidental ingestion of soil and sediment. Given differences in the behavior of different DLCs in the environment, there is some uncertainty associated with the application of a value based on TCDD to all DLCs.

An $RBA_{soil\text{-}sediment}$ of 0.50 was also adopted for assessment of exposures to arsenic via direct incidental ingestion of soil and sediment. This value was based on the findings of two meta-analyses (USEPA 2010f; Roberts et al. 2007) that reported ranges of bioavailability in soil from 0.05 to 0.31 and from 0.10 to 0.61, respectively. These meta-analyses are summarized below:

- USEPA (2010f) completed in vivo tests of 29 test materials from contaminated arsenic and clean sites using the Juvenile Swine Model. The test materials represented a variety of arsenic phases (e.g., oxides, sulfates, phosphates). Discounting three tests that were determined to be unreliable due to levels of administered arsenic, estimated bioavailability values ranged from less than 0.10 to 0.61 (i.e., 10 to 61 percent) with a mean of 0.34 (i.e., 34 percent). Based on these findings USEPA Region 8 concluded that a RBA of 0.50 as a generally conservative default value for inorganic arsenic (USEPA 2012a).
- Bioavailability studies conducted by Roberts et al. (2007) in cynomolgus monkeys measured the bioavailability of arsenic in 14 soil samples from 12 different sites, including mining and smelting sites, pesticide facilities, cattle dip vat soil, and chemical plant soil. The reported bioavailability ranged from 0.05 to 0.31 (i.e., 5 to 31 percent).

Based on the above studies, the term "absorbed fraction from exposure medium on site" in Equation 5-5 was conservatively assumed to be 0.50. The absorbed fraction from drinking water, which is the dosing medium in the study that provides the basis for the toxicity criteria for inorganic arsenic used for this BHHRA, was assumed to be 1. Therefore the $RBA_{soil\text{-}sediment}$ for arsenic was set to 0.50 for the BHHRA.

A $RBA_{soil\text{-}sediment}$ for all other $COPC_{HS}$ was conservatively assumed to be 1.0. Additionally, the relative bioavailability from tissue ingestion (RBA_{tissue}) was assumed as 1.0 for all $COPC_{HS}$.

Dermal Absorption Factor for Soil and Sediment

The dermal absorption factor represents the proportion of a chemical that is absorbed across the skin from the soil and/or sediment matrix once it has been contacted. Skin permeability is related to the solubility or strength of binding of the chemical in the soil or sediment matrix compared to the skin's stratum corneum and the degree to which the chemical can penetrate the stratum corneum to enter the bloodstream. Therefore, dermal absorption is dependent on the properties of the chemical itself, as well as on external factors including the physical properties of the soil or sediment matrix (e.g., particle size, organic carbon content) and the conditions of the skin (e.g., skin condition, moisture content). Data with which to characterize dermal absorption of chemicals from sediment is not readily available and dermal absorption of chemicals from soil and sediment matrices will differ to some degree. In the absence of sediment-specific information, however, USEPA (2004) supports the application of factors derived for soil to sediment.

Dermal absorption factors for dioxins and furans, arsenic, PCBs, and bis(2-ethylhexyl)-phthalate (BEHP) were obtained from USEPA (2004). Those for chromium, mercury, and nickel were obtained from the California Environmental Protection Agency, Office of Environmental Health Hazard Assessment's (OEHHA) *Technical Support Document for Exposure Assessment and Stochastic Analysis, Draft* (CalEPA 2011). Following USEPA (2004) guidance, in the absence of available data for copper and zinc, a conservative dermal absorption factor of 1.0 was assumed for these $COPC_{HS}$. The dermal absorption factors applied in this BHHRA are presented in Table 5-7.

Chemical Reduction Due to Preparation and Cooking

It is well recognized that preparation and cooking may reduce chemical concentrations in fish tissues, particularly for lipophilic compounds such as dioxins, furans, and PCBs (USEPA 2000b, 2002b; Wilson et al. 1998). These changes are dependent on a number of factors including the lipophilicity of the compound, the type of fish, and the parts of the fish consumed.

For the deterministic CTE and RME evaluations, a cooking loss of 0 (zero percent loss) was conservatively assumed for PCBs and dioxins. In line with the EAM (Appendix A), the impact of applying a cooking loss of 0.25 (25 percent loss) was explored in the uncertainty evaluation for the risk characterization and available information on distributions of cooking loss were considered in the PRA. Following the submittal of the EAM in May 2012, a meta-analysis was published that provided a critical review of the available data on cooking loss factors for lipophilic compounds (AECOM 2012). The findings of this study are also discussed in the uncertainty evaluation.

5.1.2.3 Probabilistic Exposure Evaluation

A probabilistic exposure evaluation was completed for scenarios that met one or more of the following thresholds (Figure 1-4):

(1) The cumulative estimated exposure from all pathways resulted in an incremental cancer risk $>1\times10^{-4}$
(2) The cumulative estimated exposure from all pathways resulted in a total endpoint-specific noncancer HI>1
(3) The cumulative estimated exposure from all pathways resulted in a dioxin cancer HI>1.

The PRA focused on chemicals that were identified as potential risk drivers. Risk drivers were defined as $COPC_{HS}$ that contributed at least five percent of overall risk or hazard across all exposure pathways that made up the selected scenario, and contributed more than 5 percent to the pathway-specific risk or hazard associated with the medium of interest. Both potential exposures within USEPA's Preliminary Site Perimeter and background exposures were evaluated.

Based on the thresholds described above, a PRA was completed for a hypothetical young child fisher and a hypothetical young child recreational visitor. A single model was used to evaluate all hypothetical fishers (i.e., recreational and subsistence). The selection of these receptor groups, as well as the specific scenarios evaluated, are described further in Section 5.2.3.3 of the risk characterization. The general methods, EPCs, and exposure parameters used in the PRA are presented below, with supporting materials provided in Appendix G.

5.1.2.4 General Methods

Probabilistic analyses were completed using Oracle® Crystal Ball software (Gentry et al. 2005). Crystal Ball employs Monte Carlo analysis, a commonly used probabilistic numerical technique where the uncertainty and variability in exposure and resulting hazard/risk estimates are characterized by developing distributions that present the full range of potential exposures.